ER9004

VALUING CHANGES IN DRINKING WATER QUALITY USING AVERTING EXPENDITURES

Charles W. Abdalla
Assistant Professor of Agricultural Economics

Donald J. Epp
Professor of Agricultural Economics

Brian A. Roach
Graduate Research Assistant
in Agricultural Economics

Prepared for
United States Environmental Protection Agency
Cooperative Agreement No. CR-815412-01-0

February 1990

PENNSTATE

ENVIRONMENTAL RESOURCES
RESEARCH INSTITUTE

UNIVERSITY PARK, PA

DISCLAIMER

VALUING CHANGES IN DRINKING WATER QUALITY USING AVERTING EXPENDITURES

ABSTRACT

Drinking water quality is an important public policy issue in the United States. Analysis of drinking water policies include economic valuation of the costs which contamination imposes upon society. An important impact of drinking water contamination is an increase in the health risk of water use. Individuals affected by drinking water contamination may take measures to reduce or eliminate the risk. These measures are referred to as averting behaviors. This study focuses on the usefulness of averting expenditures in estimating the value of changes in drinking water quality.

Mathematical and graphical models interpret averting behavior costs as a lower-bound estimate of the true willingness to pay for marginal and non-marginal changes in drinking water quality. Empirical analysis indicates that household level averting expenditures increased by 58% when a community was affected by drinking water contamination. Factors influencing these averting costs included the presence of young children in the household, perceptions of the health risk of the contaminant and the amount of information received or obtained about the contaminant. Also, the implications of the findings for academic researchers and policy-makers are discussed.

ACKNOWLEDGEMENTS

The authors wish to acknowledge the useful input and comments from Dr. James S. Shortle and Dr. Cathy A. Hamlett of the Department of Agricultural Economics and Rural Sociology, Penn State University. Dr. William B. O'Neil, Office of Policy, Planning and Evaluation, US EPA, and Dr. Alan J. Krupnick, Resources for the Future, also provided suggestions which aided in planning and conducting the research.

TABLE OF CONTENTS

	Page
LIST OF FIGURES	iv
LIST OF TABLES	v
Chapter 1. INTRODUCTION	1
Chapter 2. METHODS OF ECONOMIC BENEFIT ESTIMATION	5
2.1. Introduction	5
2.2. Willingness to Pay	5
2.3. Definition of the Averting Behavior Method	6
2.4. Alternative Methods of Benefit Estimation	7
2.4.1. The Cost of Illness Method	7
2.4.2. The Contingent Valuation Method	8
2.5. Theoretical Development of the Averting Behavior Method	8
2.6. Empirical Averting Behavior Studies	13
2.7. Summary of Pilot Study	14
2.8. Model Construction for Marginal Changes in Water Supply Quality	16
2.9. Analyzing Averting Expenditure Levels for Non-Marginal Changes in Water Quality	23
2.10. Possible Complications and Further Considerations of the Model	32
2.11. Risk Perception and Valuation	37
2.12. Summary	39
Chapter 3. PROCEDURES	41
3.1. Introduction	41
3.2. Description of Study Sites	41
3.3. Data Collection	44
3.4. Analysis Methods	45
3.5. Variable Descriptions	46
3.5.1. Dependent Variables	46
3.5.2. Independent Variables Included in Both Analyses	46
3.5.3. Independent Variables Included Only in the Doylestown Analysis	50
3.5.4. Independent Variables Included Only in the Perkasie Analysis	51
3.6. Risk Perception and Analysis	52
Chapter 4. DESCRIPTIVE EMPIRICAL RESULTS	55
4.1. Introduction	55
4.2. Response Rate of the Perkasie Survey	55
4.3. Descriptive Results of Perkasie Survey	55
4.4. Averting Expenditure Estimates of the Perkasie TCE Contamination	56
4.4.1. Cost of Increased Purchases of Bottled Water	58
4.4.2. Cost of New Purchases of Bottled Water	59

TABLE OF CONTENTS (cont.)

	Page
4.4.3. Cost of Home Water Purification Systems . . .	59
4.4.4. Cost of Hauling Water	60
4.4.5. Cost of Boiling Water	61
4.5. Comparison of Perkasie Mail Questionnaire Results with Telephone Interview Results	61
4.6. Total Costs of the Perkasie TCE Contamination . .	63
4.7. Response Rate of the Doylestown Survey	63
4.8. Descriptive Results of the Doylestown Survey . . .	63
4.9. Comparison of Doylestown Mail Questionnaire Results with Telephone Interview Results	66
4.10. Doylestown Sample Averting Expenditures	66
4.11. Doylestown Population Averting Expenditures . . .	67
4.12. Comparison of Demographic Variables for Perkasie and Doylestown Samples	67
4.13. Risk Analysis	69
4.13.1. Risk Perceptions of Perkasie Sample	69
4.13.2. Risk Perceptions of Doylestown Sample . . .	72
4.13.3. Comparison of Risk Perceptions for Doylestown and Perkasie	72
4.13.4. Value of a Statistical Life Derived from Perkasie TCE Contamination Incident	72
Chapter 5. REGRESSION RESULTS	74
5.1. Introduction	74
5.2. Regression Procedures	74
5.3. Doylestown Regression Results	75
5.4. Perkasie Regression Results	80
5.5. Regression Alternatives	84
5.6. Comparison of Doylestown and Perkasie Regression Results	89
Chapter 6. CONCLUSIONS AND POLICY IMPLICATIONS	91
6.1. Introduction	91
6.2. Summary of Conclusions	91
6.3. Research Limitations	92
6.4. Policy Implications	93
6.5. Suggestions for Future Research	95
REFERENCES	97
Appendix A. COPY OF QUANTITATIVE RISK ELICITATION METHOD USED FOR LEMONT PILOT STUDY	104
Appendix B. COPY OF LETTER NOTIFYING PERKASIE RESIDENTS ABOUT THE PRESENCE OF TCE IN THE WATER	106
Appendix C. COPY OF QUESTIONNAIRE SENT TO DOYLESTOWN SAMPLE	109
Appendix D. COPY OF QUESTIONNAIRE SENT TO PERKASIE SAMPLE	120
Appendix E. COPY OF REMINDER POSTCARD SENT TO PERKASIE AND DOYLESTOWN SAMPLES	133
Appendix F. COPY OF FOLLOW-UP TELEPHONE INTERVIEW FOR PERKASIE SAMPLE	135

TABLE OF CONTENTS (cont.)

		Page
Appendix G.	COPY OF FOLLOW-UP TELEPHONE INTERVIEW FOR DOYLESTOWN SAMPLE	137
Appendix H.	FREQUENCY OF PERKASIE VARIABLES NOT PRESENTED IN TEXT	139
Appendix I.	FREQUENCY OF DOYLESTOWN VARIABLES NOT PRESENTED IN TEXT	152
Appendix J.	SUMMARY OF TWO-STAGE LEAST SQUARES APPLICATION.	162

LIST OF FIGURES

		Page
Figure 2.1	Health Cost Function	19
Figure 2.2	Consumption Possibilities Curve	24
Figure 2.3	Compensating and Equivalent Variation	25
Figure 2.4	Compensating Variation ($\Delta H=0$)	27
Figure 2.5	Compensating Variation ($\Delta X=0$)	29
Figure 2.6	Compensating Variation ($\Delta X=\|CV\|$)	30
Figure 2.7	Equivalent Variation ($\Delta H=0$)	33
Figure 2.8	Equivalent Variation ($\Delta X=0$)	34
Figure 2.9	Compensating Variation = Equivalent Variation .	35
Figure 3.1	Location of Perkasie and Doylestown	42

LIST OF TABLES

		Page
Table 2.1	Summary of Obtained Values of a Statistical Life (Viscusi, 1986)	40
Table 4.1	Frequency of Trips to Haul Water for the Sole Purpose of Hauling Water During Perkasie TCE Contamination	57
Table 4.2	Estimated Costs of the Perkasie TCE Contamination Incurred by Perkasie Sample . . .	64
Table 4.3	Estimated Costs of the Perkasie TCE Contamination Incurred by Perkasie Population .	65
Table 4.4	Comparison of Independent Variables of Doylestown and Perkasie Samples	68
Table 4.5	Qualitative Rating of Cancer Risk of Perkasie Water Prior to TCE Contamination . . .	70
Table 4.6	Qualitative Rating of Cancer Risk of Perkasie Water Associated with the TCE Contamination	71
Table 4.7	Qualitative Rating of Cancer Risk Associated with Doylestown Water	72a
Table 5.1	Pearson Correlation Coefficients of Independent Variables to Doylestown Averting Expenditures .	77
Table 5.2	Regression Results from Doylestown Sample (Dependent Variable - Estimated Yearly Averting Expenditures)	78
Table 5.3	Pearson Correlation Coefficients of Independent Variables to Natural Logarithm of Doylestown Averting Expenditures	79
Table 5.4	Regression Results from Doylestown Sample (Dependent Variable - Natural Logarithm of Estimated Yearly Averting Expenditures)	81
Table 5.5	Pearson Correlation Coefficients of Independent Variables to Perkasie Averting Expenditure Increase Associated with TCE Contamination . . .	82

LIST OF TABLES (Cont.)

Page

Table 5.6 Perkasie Regression Results (Dependent Variable - Estimated Household Averting Expenditures Resulting from TCE Contamination) . 83

Table 5.7 Perkasie Regression Results (Dependent Variable - Estimated Household Averting Expenditures Resulting from TCE Contamination, Only Significant Variables) 85

Table 5.8 Pearson Correlation Coefficients of Independent Variables to Natural Logarithm of Perkasie Averting Expenditure Increase Associated with TCE Contamination 86

Table 5.9 Perkasie Regression Results (Dependent Variable - Natural Logarithm of Estimated Household Averting Expenditures Resulting from TCE Contamination 87

Table 5.10 Perkasie Regression Results (Dependent Variable - Natural Logarithm of Estimated Household Averting Expenditures Resulting from TCE Contamination, Only Significant Variables 88

Chapter 1

INTRODUCTION

Pure and reliable water is necessary for sustaining all life. However, the quality of water supplies is often threatened by contamination from many different sources (U.S. Congress, 1987). Due to increasing public awareness and concern, water quality has become an important public policy issue in the United States. Considerable legislative effort is being devoted to water quality issues in an attempt to protect the nation's water supplies. These issues include the accurate monitoring of the quality of water supplies and setting efficient water quality standards (U.S. Congress, December 1985).

Approximately fifty percent of the water used for residential purposes in the United States is obtained from groundwater sources (Solley, et al., 1985). Groundwater quality is often threatened from agricultural, industrial and municipal activities. Groundwater contamination incidents are occurring with increasing frequency in the United States. Guidelines and standards for groundwater quality were established by thirty-seven states between 1985 and 1987 (U.S. EPA, 1988). Groundwater sources are being monitored more closely to detect the presence of hazardous contaminants.

The effective management of groundwater resources requires intelligent policy-making strategies (McDonald and Kay, 1988). Policies are established utilizing various inputs into the decision-making process, including economic, legal, political and administrative (Greenley, et al., 1982). These sources can provide decision makers with guidelines to follow in determining which policies to adopt.

Economics presents many such guidelines for establishing efficient policies. Most economic policy analysis is based on some form of social welfare function to weigh costs and benefits. Accepted theory suggests that policies should be implemented if the economic benefits of the policy are greater than the economic costs, unless there exists an alternate policy which is clearly more beneficial (Mills, 1978). Such a policy is considered to increase the welfare of society as a whole.

The costs of implementing policies which improve the quality of some environmental factor can normally be estimated directly. For example, improving the quality of water supplied to residences may involve installing better purification systems in municipal plants, more accurate monitoring systems and other equipment. While estimating costs may be a complicated task, few difficulties arise identifying and determining costs. However, since a market does not exist for environmental quality, the benefits of environmental improvement policies are difficult to quantify in economic terms (Freeman, 1979).

Several techniques are available to estimate the economic benefits of environmental improvements or, conversely, the economic costs of detrimental environmental changes. These approaches are either indirect, normally using existing markets to infer a value towards a non-market commodity, or direct, presenting a hypothetical market where valuations are expressed. The three estimation methods most frequently utilized when analyzing environmental issues affecting human health are the cost of illness, contingent valuation and averting behavior methods (Dickie and Gerking, 1988). Each of these methods have theoretical and empirical strengths and weaknesses.

This study focuses on the applicability of the averting behavior method to the economic valuation of drinking water quality. The averting behavior method is an indirect approach to valuing changes in environmental quality. The research has the following six objectives.

The first objective is to interpret the averting behavior as a theoretical measure of the economic benefits of environmental improvement. Previous literature will be reviewed and used as a basis to develop theoretical models. These models will first consider marginal changes in drinking water quality and then be expanded to include non-marginal changes.

The second objective is to compare the averting behavior method to other economic estimation methods. Existing literature will be reviewed to compare the strengths and weaknesses of each method.

Third, the research will empirically estimate household level averting expenditures in two similar communities, one affected by drinking water contamination problems and one that is unaffected. The household costs of a water contamination incident will be calculated. Information on the economic damages of drinking water contamination can assist policy-makers to make better decisions.

Fourth, an alternate method of eliciting risk perceptions about water contamination from households will be developed. Several difficulties have been evident in recent studies of risk perceptions. Information concerning risk perceptions will be collected from available literature and previous research to determine the weaknesses of present methods for eliciting risk perceptions. The method will then be utilized in mail questionnaires to collect data on risk perceptions. Attitudes towards risks and the accuracy of risk perceptions will be analyzed by comparing objective risk with subjective risk values. The new method may be useful to researchers in developing survey instruments which can be better understood by respondents to yield more accurate and meaningful results.

The fifth objective is to determine the factors influencing averting decisions. Individuals will be asked questions via mail questionnaires concerning averting activities, risk perceptions and attitudes and demographic information. Hypotheses will be made concerning the direction of influence of independent variables on averting expenditures. Regression models will be constructed to test these hypotheses. This information is useful to policy makers since some of the independent variables, such as risk perception, can be affected by legislation while others factors are independent of legislation. The ability of policy makers to influence averting behavior depends on which factors are involved in individuals' averting decisions.

The last objective is to estimate the willingness to pay for reductions in the health risks of contaminated drinking water. Averting expenditure data will be used with risk perception information to obtain these results. The "value of a statistical life" can be calculated for the data set and compared with other studies. The value of a statistical life is an approach to determine society's willingness to pay for risk reductions. Details of this approach are presented in Chapter 3. Few empirical results are available on the value of safe drinking water. The value of a statistical life related to drinking water risks has never been calculated using household level averting expenditures. This research will present such values which can then be used to make comparisons with future studies. This information can also inform policy makers of the value individuals' assign to risk changes.

The above objectives will be met through theoretical development of models and analysis of empirical data collected from mail questionnaires. Chapter 2 introduces methods of economic benefit estimation. The theoretical foundations of the averting behavior method will be explored through the household production function. Recent studies exploring the theoretical interpretation of averting behavior estimates are reviewed. Also, the empirical conclusions of relevant articles are presented. Then, theoretical models are presented and investigated for marginal and non-marginal changes in environmental quality. The interpretations of the models are given along with possible complications affecting the accuracy and policy relevance of the averting behavior method.

Chapter 3 presents the empirical procedures. The experiment sites are introduced and explanations given for site selection. The analytical procedures are described. The model's variables are introduced, along with reasons for inclusion and the expected relationships. Lastly, the method of eliciting risk perception through a mail questionnaire is explained. The rationale supporting this method will be stated by comparing it with existing methods of determining risk.

In Chapter 4, the descriptive empirical results are presented. The averting expenditure estimates obtained from the mail questionnaires are presented. The results of risk perception are calculated and explained. Finally, the value of a statistical life is calculated using averting expenditures and compared with the findings of other researchers.

Chapter 5 presents the empirical regression results. The factors that influence averting behavior are identified and interpreted. This information can be relevant to policy-makers since some factors affecting averting behavior may be influenced by policy decisions and others may not.

Chapter 6 draws conclusions from the study's findings and presents policy implications. The useful information relevant to both academic researchers and policy makers will be summarized. Conclusions will be made concerning the strengths, weaknesses and applicability of the averting behavior method for drinking water quality issues. The potential strengths and limitations of the risk elicitation method used for this study are also discussed. Lastly, suggestions are made concerning future theoretical and empirical research.

Chapter 2

METHODS OF ECONOMIC BENEFIT ESTIMATION

2.1. Introduction

This chapter reviews literature relevant to the estimation of the costs and benefits associated with environmental quality changes. The first section introduces the economic measures used to quantify these costs and benefits. Then, the averting behavior valuation method is defined. The two most commonly used alternatives to the averting behavior method, cost of illness and contingent valuation, are discussed. The theoretical development of the averting behavior method is reviewed, including the household production function literature. Recent theoretical and empirical studies are presented with an emphasis on the interpretation of averting behavior results and comparisons between alternate estimation methods. Theoretical models will be developed to relate averting behavior estimates to the actual costs and benefits of marginal and non-marginal changes in environmental quality. Lastly, risk analysis is introduced as an important consideration in valuing changes in environmental quality.

2.2. Willingness to Pay

Water pollution imposes certain costs upon society that must be estimated in monetary terms to be useful in policy analysis. An economic foundation for valuing changes in environmental quality assumes that individuals affected by a change can associate some monetary value with it (Freeman, 1979). This valuation is determined by individuals' preferences, income and information concerning the effects of a change. Note that actual, ex post, or proposed, ex ante, changes may be valued. From an individual's viewpoint, an environmental improvement can be defined as a change that directly or indirectly results in increases in utility with other factors being held constant. In most circumstances, the determination of a positive or negative environmental change is based on reasonable assumptions that define the preferences of rational individuals.

Willingness-to-pay (WTP) is the maximum amount of money affected individuals would pay for a positive environmental change. At the WTP amount, individuals are assumed to be indifferent between the present situation and the improvement with the payment. Willingness-to-accept (WTA) is defined as the minimum monetary compensation that would have to be given to affected individuals after a negative environmental change so that they would be at the same utility level as before the change. Both WTP and WTA can be stated as the necessary income change, positive or negative, that returns the affected individual to their original utility level. Estimates of WTP and WTA can be used to value marginal and non-marginal changes in environmental quality.

There are three widely accepted methods for analyzing the welfare effects associated with non-marginal changes in the quality of environmental factors (Freeman, 1979). These are ordinary consumer surplus (CS), compensating variation (CV) and equivalent variation (EV). Consumer surplus is defined as the area under the Marshallian demand curve but above the price level. Since the demand curve for water quality or health is difficult to determine, CS is not recognized as practical for this situation. In addition, Freeman states that CS can not be defined in terms of the underlying indifference curves and is normally different from the other measures.

Compensating variation is defined as the amount of money that can be taken away after an economic change such that the individual is no better or worse off than before. Note that CV uses the original utility level as a reference point. Equivalent variation is defined as the amount of money that would have to given to an individual, before an economic change, such that the individual is no better or worse off than if the change took place. Thus, EV and CV differ in the reference utility level. CV and EV will not necessarily be the same value but, under most circumstances, CV and EV will differ only marginally and the choice of the applicable method is often subjective (Mishan, 1981b). Graphical analyses will be used to further differentiate between CV and EV later in this chapter.

2.3. Definition of the Averting Behavior Method

Individuals exposed to an undesirable level of some environmental factor can react in several ways (Shibata and Winrich, 1983). They may choose to do nothing in response to the situation. They may attempt to improve the situation by bribing the producer of the externality or promoting public action that reduces the level of contamination (Zeckhauser and Fisher, 1976). Finally, they may take some actions to reduce or eliminate the harmful effects of the externality as a means of self-protection (Shogren and Crocker, 1989). This is known as averting behavior or averting actions.

For averting behavior (AB) to be an effective option, either market goods or protective actions must exist which reduce or eliminate the harmful effects of a contaminant. Individuals faced with this option must decide how much, if any, averting behavior will be undertaken. There is an economic cost associated with any level of averting behavior (Watson and Jakesch, 1982). This cost includes expenditures on averting goods purchased in markets and the opportunity cost of household resources allocated to averting activities. The AB method of economic estimation utilizes market and non-market information to indirectly infer values for different environmental states.

There is a difference between averting behaviors and averting expenditures. Averting behaviors refer to the actual activities undertaken by an individual for protection against the harmful effects of a contaminant. For example, using bottled water is an averting behavior which reduces exposure to contaminants in drinking water. Averting expenditures refer to the total dollar value associated with the averting behaviors being undertaken. The terms defensive behaviors and defensive expenditures can be substituted for averting behaviors and defensive expenditures respectively.

Theoretical analyses, presented later in this chapter, have developed models which indicate that AB information can estimate the economic damages associated with contamination. These estimates can provide benefit measures for environmental quality improvements. However, the applicability of AB measures is limited by the validity of the model's assumptions (Courant and Porter, 1981; Bartik, 1988).

2.4. Alternative Methods of Benefit Estimation

Several economic methods are commonly used to estimate benefits of environmental improvements. These methods include travel cost, cost of illness, hedonic pricing, contingent valuation and averting behavior. However, only the averting behavior (AB), contingent valuation (CV) and cost of illness (COI) methods have been frequently used to value changes in environmental factors which affect individuals' health (Dickie and Gerking, 1989). This section will introduce the CV and COI methods and discuss their strengths and weaknesses.

2.4.1. The Cost of Illness Method

The cost of illness method, similar to the AB method, is an indirect approach to benefit estimation which infers willingness to pay for improved environmental states. The COI method assumes that the harmful environmental factor may result in sickness or injury that requires direct expenditures for treatment (such as doctor visits or medication) and indirect costs (such as lost work and leisure time) (Harrington and Portney, 1987). Costs may also include some preventative measures, such as diagnostic expenses and immunization. Also, COI attempts to account for mortality losses and morbidity losses by estimating lost income over a lifetime. However, the COI method does not take into consideration other consequences of disease, such as pain and suffering (Cooper and Rice, 1976) . The cost to society of a negative externality is estimated by adding direct and indirect illness costs. The reduction of a negative externality level is assumed to result in reduced incidence or severity of the disease, or diseases, associated with the externality (Berger, et al., 1987) .

The appeal of the COI method appears to stem from the ease of calculation of estimates since information on every case does not need to be collected. First, the prevalence of a disease must be known in the study area. Then, an estimate is made from medical information concerning the probability that a certain environmental factor caused onset of the disease. Lastly, information is obtained from actual cases of the disease or medical records about the costs associated with these cases. The main weakness of the COI method is that theoretical analyses have found no direct connection between COI estimates and willingness to pay (Dickie and Gerking, 1989). Thus, the accuracy of COI measures in estimating willingness to pay is limited by the interpretation of the estimates. Also, most COI studies do not include information on defensive expenditures. Most researchers conclude that the COI method understates true willingness to pay (Harrington and Portney, 1987).

2.4.2. The Contingent Valuation Method

The CV method has been used by economists to value goods for which no direct market exists (Randall, et al., 1983). CV usually involves presenting a sample of individuals with a hypothetical situation in which a change affects the individual's utility. If the change is beneficial, respondents are asked how much they would be willing to pay to assure that the change occurs. If the change is detrimental, the respondents are asked how much they would be willing to pay so that the change does not take place. The hypothetical situation can be presented to the respondents through mail questionnaires or telephone or personal interviews. The reliability of the CV method depends on the realism of the situation, the wording of questions and the amount of attention the respondents give to the questions (Dickie and Gerking, 1989).

The CV method's main advantage is that it directly obtains a theoretically correct measure of willingness to pay. While the COI and AB methods are indirect approaches, CV attempts to measure willingness to pay directly by asking individuals using a hypothetical situation. However, several biases have been found to be associated with the CV method, especially when used to value environmental factors. Burness, et al. (1983) state that theorizing an environmental situation as a "commodity" that can be described in a CV market may be incorrect. Also, they question the validity of hypothetical valuation in revealing preferences. Since the individual does not have to actually pay any money in a hypothetical situation, bids for environmental improvement can be biased due to inability to consider actual income, strategic bids in which the respondent attempts to bias a bid in the direction of a desired end and misunderstanding of questions. Several researchers (Thomson 1987; Viscusi, 1986) report that many respondents have difficulties responding thoughtfully to hypothetical questions concerning health risks. Randall, et al. (1983) suggest CV surveys may contain a "hypothetical bias". This bias exists since there is no penalty for inaccurate responses. Thus, less thought and effort may be applied to the question than would exist in an actual market. Research does not suggest how much of a distortion this bias creates or even its direction. There is strong support for the contingent valuation method but many problems exist in determining the accuracy and interpretation of empirical results (Dickie and Gerking, 1989).

2.5. Theoretical Development of the Averting Behavior Method

While the importance of averting behavior in calculating the economic impacts of environmental policies has long been recognized, detailed analysis has been ongoing for about fifteen years. Early theoretical articles that analyzed environmental policy making noted that failure to consider averting possibilities when designing policies could result in inefficient outcomes. The classic article by Cease (1960) introduced the concept of bargaining between the producer of an externality and the individual adversely affected by the externality. By bargaining, the producer and individual could achieve an economically efficient solution. Cease presented an example in which a farmer's crops were being damaged by stray cattle from an adjoining ranch. The possibility was noted of erecting a fence between the properties when the rancher's herd reached a certain number of cattle. Cease concluded that the outcome would be the same regardless of how the property rights were distributed. Whether the fence would be erected would be determined by comparing the marginal benefits of the fence with the marginal costs, irrespective of the property rights.

Michelson and Tourin (1966), in deriving estimates for the costs of air pollution, indicated that defensive actions were considered in air pollution studies as early as 1913. Included in their estimates were such costs as window cleaning, house painting and dry cleaning costs. Baumol (1972) noted that the presence of averting possibilities can affect the efficient Pigouvian tax structure.[1]

The present theory of averting behavior has its conceptual foundations in household production analysis (Dickie and Gerking, 1989). Early work on household production theory includes Becker (1965) and Lancaster (1966). Becker, in developing a model for the allocation of time, departed from traditional economic theory by considering that market goods and time could be combined to produce commodities that yield direct utility. Lancaster stated that a good may be utilized by the household to produce more than one commodity and that a commodity may require several market goods as inputs. Thus, household production theory is similar to the macroeconomic theory of firms, emphasizing that inputs are used to produce a commodity according to a production function. The inputs should be utilized to achieve a least cost production of a certain quantity of the commodity. Bockstael and McConnell (1983) refer to these results as "commodity service flows". Households produce these commodity service flows according to production functions which differ across households.

An application of the household Production function has been towards health issues. In this framework, individuals are endowed with a given stock of health which diminishes with age and can be increased with investment. Grossman (1972) presented the first conceptualization of the household production of health He assumed that individuals derive utility from the flow of health and the amount of other commodities in each time

----

1 A Pigouvian tax structure involves the use of taxes or subsidies levied on externalities to induce them to produce at Pareto efficient levels (marginal cost of additional production is equal to marginal benefits) (Mishan, 1976)

period being considered. In this model, individuals have a downward sloping demand curve for health state and face an infinitely elastic (horizontal) supply curve since the cost of market goods does not vary with an individual's health stock. One simple equilibrium condition that resulted was that individuals maximize utility by setting the present value of the marginal cost of health equal to the present value of the marginal benefits. Also, the analysis demonstrated that individuals produce an efficient level of health at least cost, considering the two inputs as market goods and time, by setting the increase in health from an additional dollar of market goods equal to the increase in health from an additional dollar of time. Some insights the model provided were that the quantity of health-related market goods demanded should decline throughout one's life cycle, the demand for health stock and market goods should be positively related to the wage rate and, if education affects the production function for health, then education should increase the demand for health.

Pollack and Wachter (1975) noted that the applicability of the household production function towards the demand for health is strongly dependent upon the simplifying assumptions made. A complication that may arise is the utilization of time in the production of a commodity. Since time given to the production of many commodities may also directly enter the utility function, the price of health may be biased. If the assumptions allow the household production framework to be applied, some possibilities are suggested. First, differences among individuals in the consumption of market goods can be attributed to varying production functions instead of differences in tastes and preferences. Second, the household production function provides a model for the allocation of goods and time within the household. Lastly, the household production function concentrates on commodities while conventional demand theory deals only with market goods.

In studying the demand for wildlife recreation, Bockstael and McConnell (1980) suggested that the production of commodities is a two-step process. The first involves the determination of the least cost bundle of inputs that will produce a given level of the commodity. The second step is to maximize utility subject to an income constraint. A change in policy will reflect a change in the marginal cost of the commodity which will alter the utility maximizing solution. Later work by Bockstael and McConnell (1983) suggested that knowledge of the household production function is not necessary to value changes in welfare but that benefits and costs of policies can be valued by analyzing changes in the demand for market goods. Thus, changes in environmental quality can be studied indirectly through Marshallian demand curves.[2] The critical

----

[2] Marshallian demand curves are derived from observable demand and allow utility to change with price and income changes. Marshallian demand curves differ from Hicksian, or compensated, demand curves which keep utility constant with price and income changes and are not observable (Varian, 1978).

assumption noted by Bockstael and McConnell was that the inputs have no value other than in the production of the commodity.

The first conceptual analysis of averting behavior was made by Zeckhauser and Fisher (1976). This study concluded that a failure to include averting possibilities in designing environmental policies generally results in an underestimate of the benefits of reducing the level of a negative externality or an overestimate of the costs of increasing the level of an externality. Thus, policy-making would be biased towards the status quo situation if averting expenditures are not recognized. The article extended the example of the farmer and the rancher introduced by Cease (1960) by noting that the fence between the properties can be of different strengths. In this situation, an increase in the number of cattle in the rancher’s herd may cause the farmer to increase the strength of an existing fence. However, this does not change the efficiency conditions that were presented by Cease. Zeckhauser and Fisher presented a basic model which suggests the efficient averting behavior level is determined when the marginal cost of averting behavior is equal to its marginal benefit. Also, the consideration of moving to a new location as an averting behavior was introduced in an algorithm that determined efficient relocation of individuals when exposed to a change in the level of noise pollution.

Other researchers (Mills, 1978; Seneca and Taussig, 1979) suggested that averting expenditures may provide a lower bound estimate of the costs of pollution. Courant and Porter (1981) presented the first theoretical comparison between averting expenditure measures and willingness to pay. They presented two models dealing with changes air pollution levels, one in which air quality does not enter into the direct utility function and one in which it does. Courant and Porter showed that averting expenditures are an upper or lower bound estimate of willingness to pay depending on the assumptions about the household production function’s properties and whether air pollution directly entered the utility function. Also, they concluded that averting expenditures may not be a good approximation of willingness to pay since some theoretical components of the change in averting expenditures are difficult to measure. When air pollution directly entered the utility function, Courant and Porter stated that, even with valid assumptions about the household technology function, whether averting expenditures place any bounds on willingness to pay may be difficult to determine. Further, utility terms can enter the expression for a change in averting expenditures which cannot be estimated without information concerning the production and utility functions.

The models presented in Courant and Porter, as well as Watson and Jaksch (1982) and Harford (1983), viewed air pollution as affecting personal or household cleanliness and do not consider any health effects. Harford refers to the “unit cost of a cleaning episode” as a function of air pollution and, possibly, the frequency of cleaning episodes. These articles report difficulties in theoretically interpreting averting expenditure measures. If these models considered the health effects of changes in pollution, averting expenditure measures would be even more difficult to interpret. Of the three, only Watson and Jaksch attempted to empirically measure the welfare gains associated with improved air quality.

Harrington, Krupnick and Spofford (1986) developed a model for the loss in social welfare associated with an increase in pollution. The expression for the true cost included terms for the direct utility losses of illnesses, lost work productivity, lost work time, lost leisure time, medical expenses and defensive expenditures. Harrington and Portney (1987) reported that the sum of changes in averting expenditures and cost of illness is likely to be a lower bound estimate to willingness to pay, assuming individuals did not increase averting expenditures in response to an decrease in pollution. If pollution has direct utility effects, the probability is even higher that the sum of the changes in averting expenditures and COI would provide a lower bound to WTP. They concluded that the possibility exists to observe true benefits in principle but obtaining the necessary information regarding market and non-market behavior normally prohibits the analysis.

Dickie and Gerking (1989) reviewed the literature on the COI, CV and AB methods. They noted that CV estimates willingness to pay based on expressed preferences while AB estimates WTP based on revealed preferences. Additionally, comparison between averting behavior and COI measures is difficult since COI reflects values for society while the AB measures reflect individual behavior. The COI method estimates the total cost to society of the ailments caused by a particular environmental factor, including lost wages, foregone tax dollars and decreased productivity. The social costs obtained from COI, however, do not attempt to correspond to willingness to pay while AB and CV can be theoretically compared to WTP. An important point was that the methods have differences in the cost of operationalization since COI does not require primary data collection while the AB and CV methods normally require eliciting such information.

Bartik (1988) agreed with others (Courant and Porter, 1981; Harford, 1983) who have suggested that theoretically correct measures of WTP could be estimated using averting expenditures if information was obtainable concerning the household's production technology. In the absence of such information, he stated that upper and lower bounds to WTP could be obtained from a defensive expenditure information for marginal and non-marginal changes in pollution. Bartik defined a lower bound estimate of the benefits of a pollution reduction as the reduction in defensive expenditures necessary to reach the originally chosen level of personal environmental quality. An upper bound to the benefits of a pollution reduction was defined as the reduction in defensive expenditures necessary to reach the personal environmental quality chosen after the pollution reduction. Bartik used measures of compensating variation (CV) and equivalent variation (EV) to analyze non-marginal changes in environmental quality. Four assumptions were considered necessary to support the policy relevance of Bartik's upper and lower bound measures. First, that defensive expenditures can perfectly substitute for changes in the externality level. Second, that defensive expenditures must require no sunk costs since this will cause the expenditure function to vary according to whether or not these costs have already been incurred. Third, that the expenditure function can be estimated using a known technological relationship. Lastly, the government must be able to influence or control the externality level. Bartik concluded that further empirical work was needed to determine the policy implications of averting behavior.

Controversy exists concerning the theoretical interpretation of averting expenditures. Some researchers have stated that AB measures are not very useful (Courant and Porter, 1981; Harford, 1983) for policy analysis while others have determined that AB results have direct policy implications Harrington and Portney, 1987; Bartik, 1988). Also, disagreement exists over how AB compares with other methods of environmental benefit estimation due to varying theoretical analyses and a lack of empirical results comparing different methods. The debate over these methods is not an attempt to determine the “correct” method but to choose, among several imperfect choices, which is the most applicable to the specific situation under study.

2.6. Empirical Averting Behavior Studies

Bhagia and Stoevener (1978) made one of the first attempts to estimate how differences in an externality level affect household behavior. They utilized existing medical information from the Portland, Oregon area. Regression analysis was conducted with medical costs as the dependent variable. The independent variables included suspended particulate level as a proxy for air pollution, meteorological data and various demographic factors. The results indicated that air pollution was not significantly related to medical expenditures. However, the authors suggested that the time lag between pollution and disease onset may be responsible for the results.

Swartz and Strand (1981) studied the effect of the finding of kepone, a suspected carcinogen, in the James River in Virginia by determining shifts in the demand for seafood. They noted that inefficient behavior by households is likely to exist during a contamination incident due to incomplete and inaccurate media information. Smith and Desvousges (1986) were the first to empirically measure differences in micro level household averting expenditures based on differences in an externality level. They surveyed two samples, one that had experienced widely publicized problems with hazardous wastes contamination and one that had not had any problems with hazardous wastes. The three categories of averting behaviors that were considered were purchasing bottled water, installing a water filter and attending public meetings to express views on hazardous wastes. Probit analysis was used to estimate the likelihood that each of these behaviors would be undertaken. Smith and Desvouges noted that income did not significantly influence averting behavior decisions but information concerning contamination levels may affect averting decisions.

Harrington, Krupnick and Spofford (1989) estimated the costs associated with an outbreak of waterborne giardiasis in Pennsylvania. Losses were considered as a result of illness and from actions to avoid exposure to the contaminated water. To elicit information on defensive behavior, a random sample of 50 individuals was contacted through telephone interviews. Upper- and lower-bound estimates of total defensive expenditures were calculated. The conclusion was made that the costs of the outbreak exceeded the cost of implementing the necessary filtration system to prevent future outbreaks. So, if the probability of future outbreaks could be estimated, a determination could be made concerning whether or not the filtration system should be installed.

Gerking and Stanley (1986) estimated willingness to pay for improved air quality in St. Louis. The dependent variable included in their analysis was whether or not a doctor was seen at least once a year. Their regression results indicated that ozone concentration had a statistically significant influence at the .01 level on the dependent variable of doctor visitation. Gerking and Stanley reported, however, that the link between the dependent variable and WTP was not supported by theory and that future studies may obtain more applicable results by considering the actions taken by households to avoid exposure to air pollution.

The empirical literature on environmental benefit estimation indicates a need for more comprehensive data collection and improved survey methods. Most of the studies determined the damages caused by air pollution, through health and soiling effects. Few articles have estimated damages resulting from water contamination incidents. These have been limited in their usefulness due to small data sets, missing information and results that lack statistical significance.

2.7. Summary of Pilot Study

The experimental methods utilized in this research were significantly influenced by information obtained as result of a pilot study undertaken prior to this study (Abdalla, 1989) conducted in early 1988. While the results of the pilot study will not be reported in detail, a summary is relevant since many possibilities for experimental improvements became evident during the pilot study. Many of these changes were incorporated into the experimental methods of this research to obtain more meaningful and valid empirical results.

The pilot study focused upon a groundwater contamination incident in Centre County, Pennsylvania. The municipal water supply of the Lemont Water Company contained levels of perchloroethylene (PCE) averaging 25 parts per billion (ppb). No maximum contaminant level was set by the EPA or the Pennsylvania Department of Environmental Resources, although the proposed standard was 5 ppb. The incident lasted from June of 1987 until late December, 1987.

An initial mail questionnaire was sent to all 1600 residential customers of the Lemont Water Company in mid-February of 1988 and focused on any changes in household behavior as a result of the PCE contamination. The initial questionnaire included a question asking respondents to rate the health risks associated with the levels of PCE in their water supply on a discrete scale with six possible answers ranging from “Insignificant

Risk" to "Very High Risk". However, no information was elicited concerning the quantitative risk perception along a continuous scale. A subsequent mailing was sent to the 1045 households that returned the initial questionnaire during Spring of 1989. This asked respondents to indicate along a risk ladder their perceived risk of the contamination incident in terms of the quantitative increase in the risk of developing cancer throughout one's lifetime. The risk ladder was presented as a linear scale ranging from a one in a million increase in the risk of developing cancer to a 100 in a million increase. A copy of this mailing is included in Appendix A.

The initial survey and follow-up was found to be insufficient in several respects after examining comments from respondents and assessing the needs of this project. First, in the initial questionnaire, respondents were asked to report behavior that had occurred at least two months previously. This delay was likely to affect the accuracy of the data since it may have been difficult for respondents to accurately recall their previous actions. The information obtained from the second mailing was even less reliable since individuals were asked to report perceptions of an incident that had occurred over one year prior to the follow-up survey. This indicated that more accurate information could be obtained if an ongoing contamination incident was studied. The primary deficiency of the pilot study was the follow-up mailing as a representation of risk perceptions. The risk ladder presented a relatively narrow range of possible responses. Many individuals indicated their risk perception above the line representing a 100 in a million risk increase. Also, there was evidence of a bias according to the response of the discrete risk rating given in the initial questionnaire. Respondents that rated the health risk as "Moderate" on the discrete question tended to quantitatively rate the risk in the middle of the risk ladder. Other individuals responded in a similar fashion, rating their quantitative risk at a point along the risk ladder correlating to their qualitative risk rating. Consequently, the average rating of the quantitative risk was biased towards the middle of the risk ladder and the empirical results were considered to contain very little validity. Lastly, some respondents indicated an inability to respond due to incomplete information or confusion over how to respond to the question.

Almost 96% of the respondents answered that they were aware of the PCE contamination. Seventy-four percent, or 773 respondents, indicated that they had changed their household activities in response to the PCE contamination. An estimate of the total economic losses due to averting behaviors resulting from the PCE contamination ranged from $137,371.81 to $160,344.02 over the six month contamination period, depending on the value assigned to lost leisure time.

Regression models were constructed from the Lemont data set in a manner similar that described in Chapter 4. The dependent variable was the estimate of household averting expenditures during the contamination period. The best statistical model concluded that the qualitative risk rating, the number of newspaper articles read concerning the contamination incident and the number of children under 3 years old had a significant positive effects on averting expenditures resulting from the PCE contamination. Education was found to have a negative influence on averting expenditure increases. This may reflect knowledge among more educated individuals that the health risks of PCE were relatively small and that averting measures were not necessary.

2.8. Model Construction for Marginal Changes in Water Quality

This section examines the theoretical relationship between WTP and averting expenditures resulting from marginal water quality changes. The WTP for reducing household water contamination, and WTA for an marginal increase in contamination, can be defined using the indirect utility function. The indirect utility function is the maximum utility achievable at given prices and income (Varian, 1978). The water contamination level also enters into the indirect utility function as an external variable. The following variables, to be used throughout this chapter, can now be introduced:

V = indirect individual utility level
I = individual income
C = the external contaminant level of the individual's water supply.
P = price vector of averting goods

Let V(P,I,C) be an individual's indirect utility function. The WTP for a reduction in C from C_0 to C_1 ($C_0 > C_1$) is formally defined as:

$$V(P, I-WTP, C_1) = V(P, I, C_0). \quad (1)$$

The WTA an increase in C from C_1 to C_0 is formally defined as:

$$V(P, I+WTA, C_0) = V(P, I, C_1). \quad (2)$$

WTP and WTA are widely accepted as the theoretically correct measures for valuing economic costs and benefits of policies. Unfortunately, no widely accepted "correct" method exists for estimating WTP and WTA, through either direct or indirect approaches.

As noted previously, the averting expenditure approach draws from the household production literature. The theory of household production is analogous to the theory of production in macroeconomic analysis. In traditional production analysis, inputs are utilized to efficiently achieve the desired output. Similarly, in the household production approach, households utilize available inputs, such as market goods and time, to obtain maximum utility. Similar to firms, households are limited by a production possibilities curve derived from available technology and resources.

In the models to be developed, households produce a state of health, also defined by Grossman (1972), Cropper (1981) and Barrington and Portney (1987). All individuals are endowed with a vector of health characteristics determined by such factors as genetics, childhood development and the presence of certain diseases or ailments. An individual's health state can be affected by various factors. Some influences are directly under the control of the individual while others are external which the individual has little control over. Individuals can purchase goods which can improve their health state through both preventative and treatment methods. Thus, health is a variable which has exogenous and endogenous influences. The model assumes that health is a stochastic variable, similar to Cropper (1981). Individuals are also assumed to obtain direct utility from health, with utility increasing as health increases.

An accepted practice in economic analysis is to define a composite of commodities as a numeraire good (Mishan, 1981b). Normally, the numeraire good is stated as all other possible goods and commodities except the particular good under analysis. This numeraire good can be utilized as a medium of exchange for measuring utility changes. The numeraire is ordinarily given a unit price to simplify analysis, assuming a constant tradeoff among the components of the numeraire good. This tradeoff can be defined in dollars or actual goods. For example, an individual is able to trade a constant amount of clothing equal to Y for Z units of food. In dollar terms, the individual can simply trade one dollar of clothing for one dollar of food at a constant rate. The numeraire good is assumed to enter directly into the individual's utility function. Further, utility is assumed to increase with increasing amounts of the numeraire good.

Some averting possibilities related to water use, such as boiling or hauling water, necessitate individuals' input of time. For example, hauling water may involve driving to a certain location. The time associated with averting actions will be estimated and assigned costs in the empirical analysis. The time required for averting activities could be used for other activities, such as work or leisure. In the model below, individuals are assumed to derive utility from their leisure time, similar to the specification of Bockstael and McConnell (1980) in which individuals derived utility from recreation.

Additional specifications of the model are now possible. The following variables can be introduced:

U = direct individual utility level
H = health state
X = the numeraire good (assigned a unit price)
D = defensive good expenditures
T = total time available (fixed)
T_d = time used for defensive actions
T_w = work time
T_l = leisure time
L = amount of leisure
R = leisure expenditures
Y_i = amount purchased of the ith defensive input
P_i = unit price of the ith defensive input

The utility function can be stated as:

$$U = U(H,X,L) \tag{3}$$

with all first partial derivatives assumed to be positive. Health is assumed to be a function of the amount of each averting good purchased and the external contaminant level. Individual health can be defined as:

$$H = (Y_1,Y_2,\ldots,Y_N,C) \tag{4}$$

where $(Y_1,Y_2,\ldots,Y_N)$ is a vector of the amounts purchased of each of N possible averting goods. Health is assumed to increase with an increase in each Y_i and decrease with an increase in C. The health cost function is illustrated in Figure 2.1.

The simplifying assumption is made that the prices of averting goods (P_i's) remain constant. This is likely to be the case since water quality changes are normally localized to a certain community and price changes would tend to result only from demand shifts involving larger areas.

The model can be developed by utilizing an individual expenditure function. Consumers are assumed to reach a given level of health at least cost. The expenditure minimization problem is usually presented as:

$$\begin{aligned} &\text{Min } D = \Sigma P_i Y_i \\ &\text{subject to } H(Y_1,Y_2,\ldots Y_N,C) = H_M \end{aligned} \tag{5}$$

where H_M is a reference health level. Freeman (1983) indicates that the solution to this problem yields the expenditure function.

In this model, a nonpurchased defensive input, household time, enters the health production function. The optimal time allocated to defensive activities enters into the utility maximization problem. Accordingly, the defensive expenditure function considered here is defined contingent upon the amount of family labor allocated to defensive activities. Thus, the cost minimization problem is expressed as:

$$\begin{aligned} &\text{Min } D = \Sigma P_i Y_i \\ &\text{subject to } H(Y_1,Y_2,\ldots Y_N,C) = H_M \\ &T_d \text{ given.} \end{aligned} \tag{6}$$

The defensive expenditure function is defined as:

$$D = \Sigma P_i Y_i^*(H,C,T_d) \tag{7}$$

where $Y_i^*(H,C,T_d)$ is the value of Y_i chosen to minimize equation (6) given H, C and T_d.

The first-order partial derivatives are assumed to have the following signs:

$$\partial D/\partial C > 0 \qquad \partial D/\partial H > 0 \qquad \partial D/\partial T_d < 0. \tag{8}$$

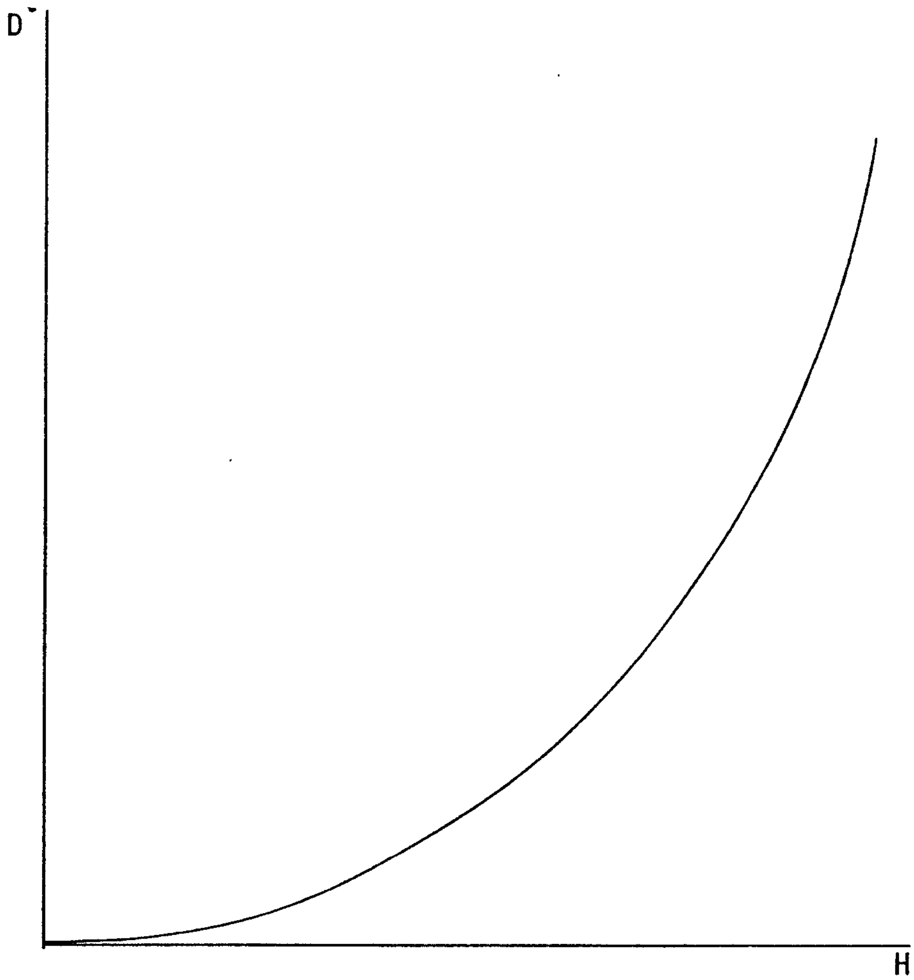

Figure 2.1 Health Cost Function

Note that actual (monetary) defensive expenditures are assumed to decrease with increasing time inputs. More time intensive averting actions, such as hauling water, are assumed to require less monetary costs than averting actions which require little time, such as purchasing bottled water or a home purification system.

The leisure expenditure function can be similarly defined. Individuals are assumed to attain a certain amount of leisure at least cost. The leisure expenditure function is stated formally as:

$$R = R(L,T_l). \tag{9}$$

The health cost function is illustrated in Figure 2.1. Health is assumed to be increasingly expensive as the health level is increased.

Income in this model is divided into two components, exogenous income, M, and income obtained from work. Individuals are assumed to work for a wage rate of W which is a function of the hours worked, T_w. Thus, total individual income in this model is defined as:

$$I = M + W(T_w) \tag{10}$$

where $w'>0$ and $w''\geq 0$.

The consumer's problem can now be expressed in terms of the expenditure functions and time and income constraints as:

$$\begin{aligned} &\text{Max } U = U(H,X,L) \\ &\text{subject to } M + W(T_w) = X + D(C,H,T_d) + R(L,T_l) \\ &\qquad \text{and } T = T_w + T_l + T_d. \end{aligned} \tag{11}$$

To solve for the optimal levels of the endogenous variables, X, L, H, T_w, T_l and T_d, a Lagrangian is set up as:

$$\begin{aligned} \text{Max } \Psi = U(H,X,L) &+ \lambda[M+W(T_w)-X-D(C,H,T_d)-R(L,T_l)] \\ &+ \delta[T-T_w-T_l-T_d]. \end{aligned} \tag{12}$$

Next, the Lagrangian is differentiated with respect to X, H, L, T_w, T_l and T_d and the results are set equal to zero to obtain the following first-order conditions:

$$\begin{aligned} &1.\ \partial\Psi/\partial H = \partial U/\partial H - \lambda(\partial D/\partial H) = 0 \\ &2.\ \partial\Psi/\partial X = \partial U/\partial X - \lambda = 0 \\ &3.\ \partial\Psi/\partial L = \partial U/\partial L - \lambda(\partial R/\partial L) = 0 \\ &4.\ \partial\Psi/\partial T_w = \lambda(\partial W/\partial T_w) - \delta = 0 \\ &5.\ \partial\Psi/\partial T_l = -\lambda(\partial R/\partial T_l) - \delta = 0 \\ &6.\ \partial\Psi/\partial T_d = -\lambda(\partial D/\partial T_d) - \delta = 0. \end{aligned} \tag{13}$$

The next step is to express the costs of an increase in C in terms of willingness to accept and willingness to pay. This can be obtained by employing the indirect utility function (Porter and Courant, 1981). The indirect utility function for this problem is defined as:

$$V = V(M,C) = U(X^*,H^*,L^*) + \lambda[M+W(T_w^*)-X^*$$
$$-D(C,H^*,T_d^*)-R(L^*,T_l^*)]$$
$$+\delta[T-T_w^*-T_l^*-T_d^*] \tag{14}$$

where X*, H*, L*, T*, T* and T* are optimal values of the choice variables. The indirect utility function is totally differentiated to obtain:

$$dV = \frac{\partial V}{\partial M}\, dM + \frac{\partial V}{\partial C}\, dC \tag{15}$$

The WTA a marginal increase in C is the increase in M such that utility remains unchanged. Accordingly, dV is set equal to zero in equation (15) and dM/dC can be stated as:

$$\frac{dM}{dC} = - \frac{\partial V/\partial C}{\partial V/\partial M}. \tag{16}$$

Equation (16) is the WTA a marginal increase in C, or equivalently, the WTP for a marginal decrease in C.

The WTA can be specified further by redefining the terms in equation (15). This can be accomplished by examining the partial derivatives of the numerator and denominator. Using equation (14), $\partial V/\partial C$ can be restated as:

$$\frac{\partial V}{\partial C} = \frac{\partial U}{\partial H^*} \frac{\partial H^*}{\partial C} + \frac{\partial U}{\partial X^*} \frac{\partial X^*}{\partial C} + \frac{\partial U}{\partial L^*} \frac{\partial L^*}{\partial C} + \lambda \frac{\partial W}{\partial T_w^*} \frac{\partial T_w^*}{\partial C}$$
$$- \lambda \frac{\partial X^*}{\partial C} - \lambda \frac{\partial D}{\partial C} - \lambda \frac{\partial D}{\partial H^*} \frac{\partial H^*}{\partial C} - \lambda \frac{\partial D}{\partial T_d^*} \frac{\partial T_d^*}{\partial C}$$
$$- \lambda \frac{\partial R}{\partial L^*} \frac{\partial L^*}{\partial C} - \lambda \frac{\partial R}{\partial T_l^*} \frac{\partial T_l^*}{\partial C} - \delta \frac{\partial T_w^*}{\partial C}$$
$$- \delta \frac{\partial T_l^*}{\partial C} - \delta \frac{\partial T_d^*}{\partial C}. \tag{17}$$

Using the first-order conditions (equation 13), most terms cancel out and the resulting expression is:

$$\frac{\partial V}{\partial C} = - \lambda \frac{\partial D}{\partial C}. \tag{18}$$

The $\partial V/\partial M$ expression is redefined by partially differentiating the indirect utility function with respect to M to obtain:

$$\frac{\partial V}{\partial M} = \frac{\partial U}{\partial H^*}\frac{\partial H^*}{\partial M} + \frac{\partial U}{\partial X^*}\frac{\partial X^*}{\partial M} + \frac{\partial U}{\partial L^*}\frac{\partial L^*}{\partial M} + \lambda + \lambda\frac{\partial W}{\partial T_w^*}\frac{\partial T_w^*}{\partial M}$$
$$- \lambda\frac{\partial X^*}{\partial M} - \lambda\frac{\partial D}{\partial H^*}\frac{\partial H^*}{\partial M} - \lambda\frac{\partial D}{\partial T_d^*}\frac{\partial T_d^*}{\partial M} - \lambda\frac{\partial R}{\partial L^*}\frac{\partial L^*}{\partial M}$$
$$- \lambda\frac{\partial R}{\partial T_l^*}\frac{\partial T_l^*}{\partial M} - \delta\frac{\partial T_w^*}{\partial M} - \delta\frac{\partial T_l^*}{\partial M} - \delta\frac{\partial T_d^*}{\partial M}. \qquad (19)$$

Using the first-order conditions from equation (13), most terms cancel out and the simplified expression is:

$$\frac{\partial V}{\partial M} = \lambda. \qquad (20)$$

Substituting the expressions obtained in equations (18) and (20) into the expression for dM/dC in equation (16) results in:

$$\frac{dM}{dC} = \frac{\partial D}{\partial C}. \qquad (21)$$

This result is the true WTA measure, or the necessary compensation to hold utility constant with an increase in the contaminant level as defined in equation (2). In other words, this is the increase in expenditures necessary to maintain the original utility level given the original optimal T_d. The sign of equation (21) is expected to be positive since the direct effect of increase in C should cause individuals to increase their averting expenditures. This is a reasonable expectation since individuals are assumed to be damaged by an increase in C. Also, contingent valuation studies have reported positive contingent bids for improvements in environmental quality (Berger, et al., 1987).

The theoretical measure of WTA will generally differ from the observed change in averting expenditures associated with an increase in the contaminant level. The actual change in D that will be estimated in this research is the change in averting expenditures plus the time costs. The change in monetary defensive expenditures is obtained by totally differentiating the defensive expenditure function, equation (6), with respect to C and is expressed as:

$$\frac{dD}{dC} = \frac{\partial D}{\partial C} + \frac{\partial D}{\partial H}\frac{\partial H^*}{\partial C} + \frac{\partial D}{\partial T_d}\frac{\partial T_d^*}{\partial C}. \qquad (22)$$

The first term of equation (22) is the true WTA from equation (21). The sign of $\partial H/\partial C$ is negative as long as health is not a Giffen good. The sign of $\partial D/\partial H$ is positive according to the cost function illustrated in Figure 2.1. Thus, the sign of the second term of equation (22) is generally negative. The sign of $\partial D/\partial T$ is demonstrated as negative by referring to condition 6 of equation (13). The equality is satisfied only when $\partial D/\partial T$ is negative, assuming that λ (the valuation of a marginal increase in income) and δ (the valuation of a marginal increase in time) are both positive. The sign of $\partial T_d/\partial C$ is assumed to be positive since individuals are hypothesized to spend more time undertaking defensive actions when contamination increases.

An additional term expressing the implied cost of additional time spent undertaking averting activities must be added to equation (22) to obtain the total value of the increase in resources allocated to pollution aversion. The change in time spent undertaking defensive actions in response to a contaminant level change is defined as $\partial T/\partial C$. The opportunity cost of time, in terms of the numeraire good, is the wage rate (see conditions 2, 4, 5 and 6 of equation 13).

Accordingly, the total value of the change in resources allocated to defensive activities (D) associated with a contamination change, including the opportunity costs ofhousehold time used in averting activities, is:

$$\frac{\partial D_0}{\partial C} = \frac{\partial D}{\partial C} + \frac{\partial D}{\partial H}\frac{\partial H}{\partial C} + \frac{\partial D}{\partial T_d}\frac{\partial T_d}{\partial C} + w\frac{\partial T_d}{\partial C} \tag{23}$$

Condition 6 of equation (13) can be utilized to redefine $\partial D/\partial T_d$ as $-(\delta/\lambda)$. Since W equals δ/λ from equation (21), the last two terms of equation (23) cancel out and the observed change in defensive expenditures simplifies to:

$$\frac{\partial D_0}{\partial C} = \frac{\partial D}{\partial C} + \frac{\partial D}{\partial H}\frac{\partial H}{\partial C} \tag{24}$$

As previously stated, the first term on the right-hand side of equation (24) is the WTA. The second term is negative. Thus, the change in defensive expenditures underestimates true WTA for an increase in C, or WTP for a C decrease. The change in defensive expenditures provides a lower-bound estimate of WTA or WTP based on the model's assumptions. This conclusion has also been reached by Barrington and Portney (1987), Berger, et al. (1987) and Bartik (1988) with similar assumptions.

2.9. Analyzing Averting Expenditure Levels for Non-Marginal Changes in Water Quality

This section analyzes the relationship between the change in defensive expenditures and the CV and EV measures of economic benefits for non-marginal changes in water quality. The difference between CV and EV can be further illustrated graphically. There exists a tradeoff between X and H according to the assumptions made in the marginal analysis. The price of X is assumed to be constant at a unit value. Health is assumed to be increasingly expensive as health increases. A consumption possibilities curve, CC, is depicted in Figure 2.2. Compensating variation and equivalent variation are illustrated in Figure 2.3. Before a contamination increase, H_0 and X_0 are the chosen optimal values to achieve a maximum utility of U_0. The increase in C causes the consumption constraint to rotate inward from point M to CC_1 with the new optimal values of the choice variables as X_1 and H_1. The CV is measured by shifting CC_1 upward in a parallel manner until it is tangent to U_0. The EV is measured by shifting CC_0 downward by parallel amounts until it is tangent to U_1.

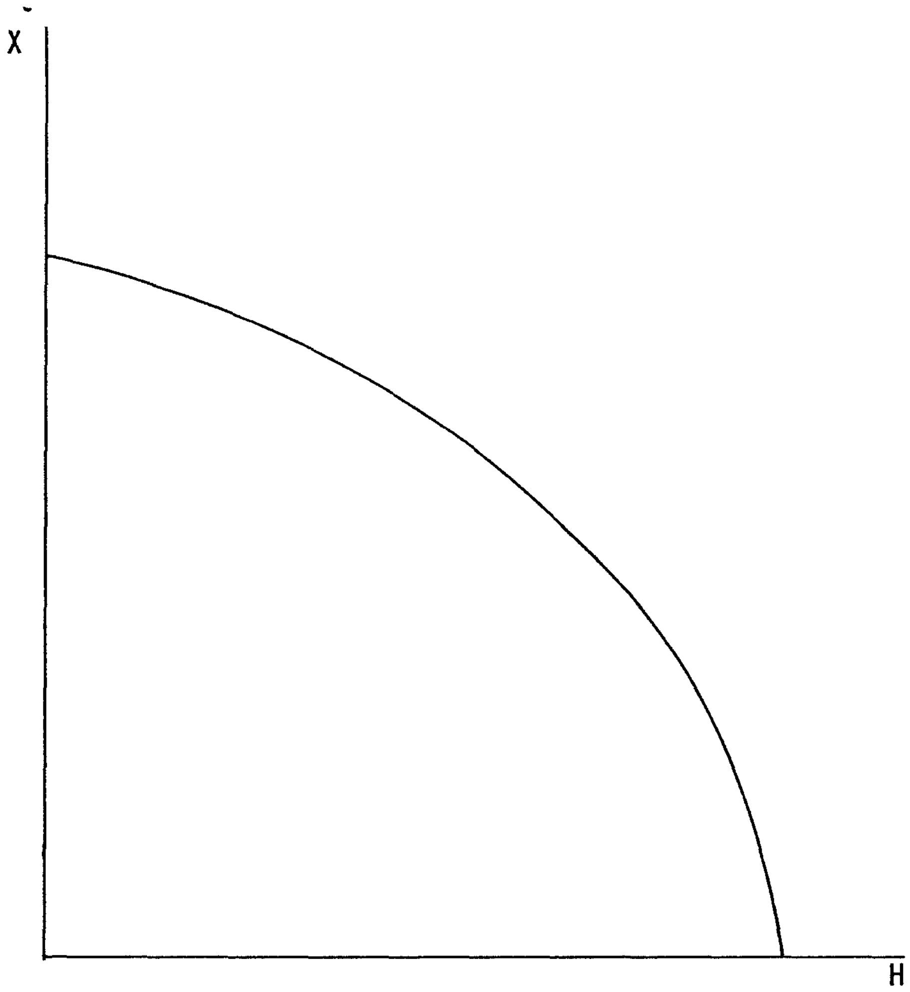

Figure 2.2 Consumption Possibilities Curve

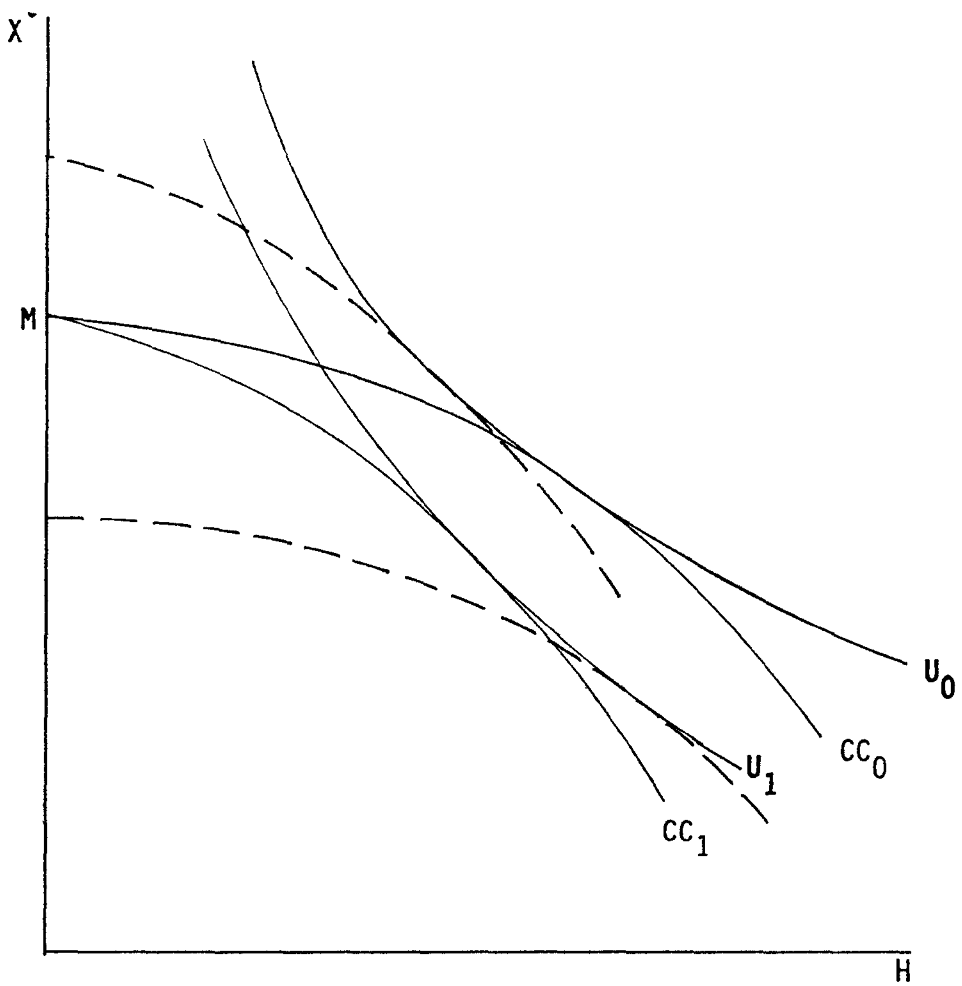

Figure 2.3 Compensating and Equivalent Variation

An expenditure function can be defined as E(C,U) where E is the expenditure necessary to reach utility U at a contaminant level of C. For an increase in C from C_0 to C_1, CV and EV can stated as:

$$CV = E(C_1,U_0) - E(C_0,U_0)$$
$$EV = E(C_1,U_1) - E(C_0,U_1). \quad (25)$$

The purpose of the following analysis is to compare the observable change in defensive expenditures with CV and EV for an increase in contamination. The actual change in defensive expenditures is:

$$\Delta D = D(H_1,C_1) - D(H_0,C_0) \quad (26)$$

where AD is assumed to be a positive value. In this model, households' expenditures must either be for X or for defensive inputs. The expression for CV in equation (25) can be expanded to:

$$CV = [D(C_1,U_0)+X(C_1,U_0)] - [D(H_0,C_0)+X(H_0,C_0)]. \quad (27)$$

Note that expenditures must be allocated between X and D in the first term of the CV expression in equation (27) so that utility is held constant at U_0 but health is not restricted to a specific value. By adding $[D(H_0, C_0)+X(H_0, C_0)]$ to equations (26) and (27) and setting the two expressions together, CV and AD can be compared using the following expressions:

$$D(C_1,U_0) + X(C_1,U_0) \text{ is } <,> \text{ or } = \text{ to}$$
$$D(H1,C1) + X(H_0,C_0). \quad (28)$$

If the left-hand side of equation (28) is greater than the right-hand side, then $CV>\Delta D$. Conversely, if the right-hand side is greater than the left-hand side, then $\Delta D>CV$. These values can be compared by utilizing graphical analyses. Several possibilities will be explored and their implications discussed.

The analyses can be simplified by making some further restrictions. First, the increase in C will not result in an increase in the health variable. In other words, health is not a Giffen good. Second, the increase in C will not result in an increase in expenditures on X. This implies that defensive expenditures will not decrease in response to the increase in C. This appears to be reasonable based on theoretical and empirical evidence (Smith and Desvouges, 1985).

The boundaries of possible outcomes can now be demonstrated. Figure 2.4 represents the restriction of the first assumption. The increase in C shifts the consumption possibilities constraint from CC_0 to CC_1. Let health remain constant at H_0 after optimizing along CC_1. Note that the change in X must equal the negative of the change in D since income is constant before any CV considerations. The change in defensive expenditures is X_0-X_1. The $|CV|$ is indicated in the graph. By definition, $|CV|$ is the vertical distance between CC_V and CC_1. The curve CC_V must lie below CC_0 at the vertical line originating at H_0 based on the assumed

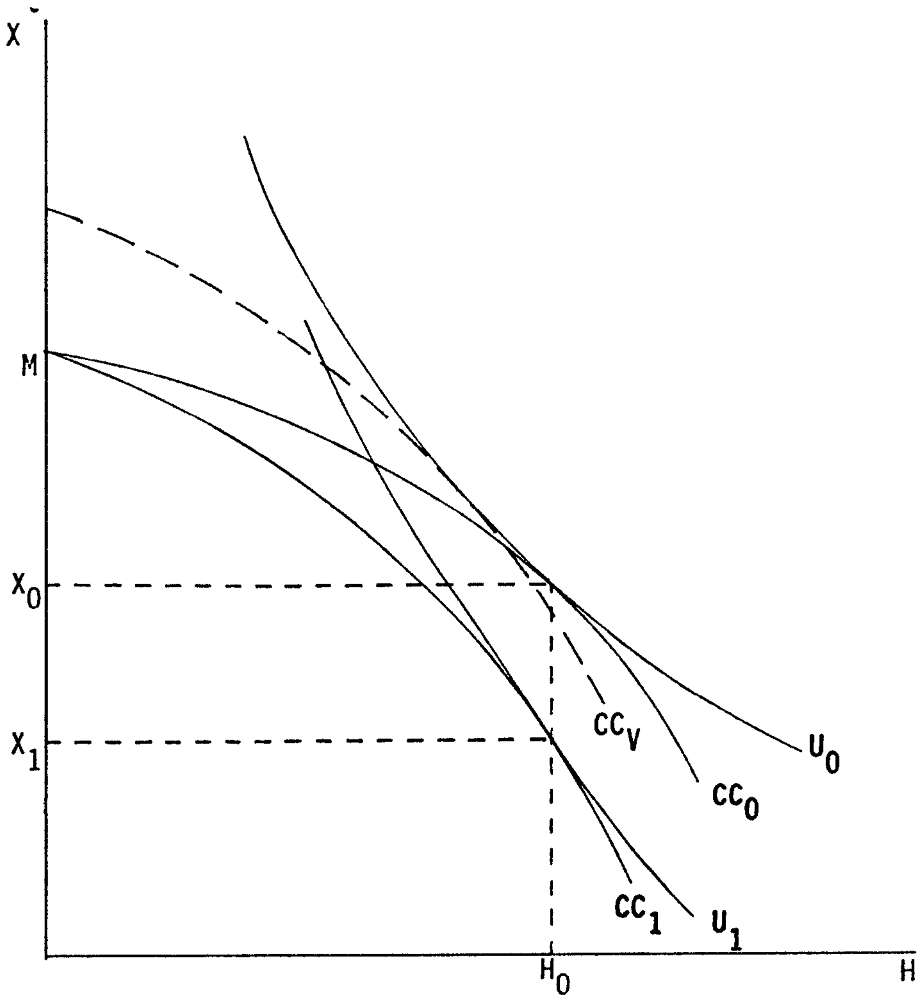

Figure 2.4 Compensating Variation (ΔH=0)

convexity of indifference curves and concavity of the consumption possibilities constraint. The slope of CC_V is greater in absolute value than CC_0 along a vertical line from H_0 since health is assumed not to increase. In this extreme situation, the conclusion is that AD is greater than $|CV|$.

The implications of the second assumption are illustrated in Figure 2.5. Exploring the limit of the assumption, let the change in C result in no change in X according to the positions of the indifference curves. Since $-\Delta X=\Delta D$, the change in D must also be zero in this case. The $|CV|$ is some positive value in the graph so $|CV|$ is clearly greater than the change in defensive expenditures when $\Delta X=0$.

The curve of possible optimized outcomes of X and H after a C increase can now be restricted to the line CC_1 below the horizontal line at X_0 and to the left of the vertical line at H_0. Since $\Delta D>|CV|$ when $\Delta H=0$ and $|CV|>\Delta D$ when $\Delta X=0$, there must be a situation where $|CV|$ =AD. This situation is represented in Figure 2.6. To obtain the condition when $|CV|=\Delta D$, equation (28) can be set as an equality. Total expenditures are limited by income without any CV considerations and CV can be described as an increase in income. Thus, the first expression of equation (28) can be rewritten as:

$$D(C_1,U_0) + X(C_1,U_0) = CV + D_1 + X_1 \tag{29}$$

where D_1 and X_1 are optimal values chosen after the increase in C.

An assumption is that X decreases with the increase in C, or that $\Delta D>0$, since a case of $|CV|=\Delta D$ is being considered. So, $X_0=X$ -AX where ΔX is negative. The second expression of equation (28) can now be stated as:

$$D(H_1,C_1) + X(H_0,CO) = D_1 + X_1 - \Delta X. \tag{30}$$

The condition for $|CV|=\Delta D$ that follows is:

$$|CV| + D_1 + X_1 = D_1 + X_1 - \Delta X. \tag{31}$$

This expression simplifies to:

$$|CV| = \Delta X. \tag{32}$$

Referring to Figure 2.6, note that X_0-X_1 is equal to $|CV|$ measured along a vertical line at H_1 since CC_V intersects a horizontal line at X_0 at point A along the vertical line at H_1. If a CV was given to a household or individual, optimization would occur where the slope of CC_V equals the slope of U_0. This would occur at point B and the resulting level of health would be H_V. The assumptions of the model dictate that $H_V<H_1$. Compensating variation can be viewed as an increase in income in this situation. The case of $H_V<H_1$ would occur only when health has a negative income elasticity. While this is a possibility, the outcome appears unlikely under reasonable assumptions. The assumption made in the marginal analysis that $\partial U/\partial H>0$ appears appropriate for non-marginal

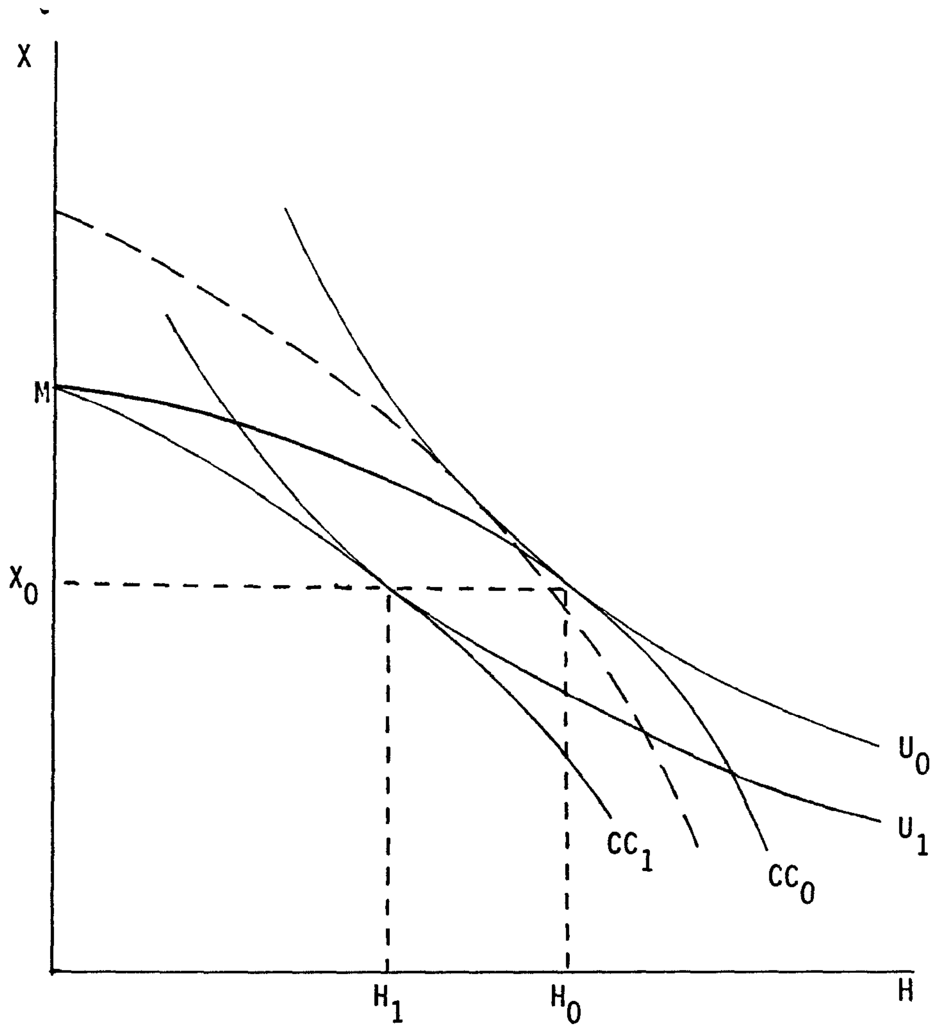

Figure 2.5 Compensating Variation ($\Delta X=0$)

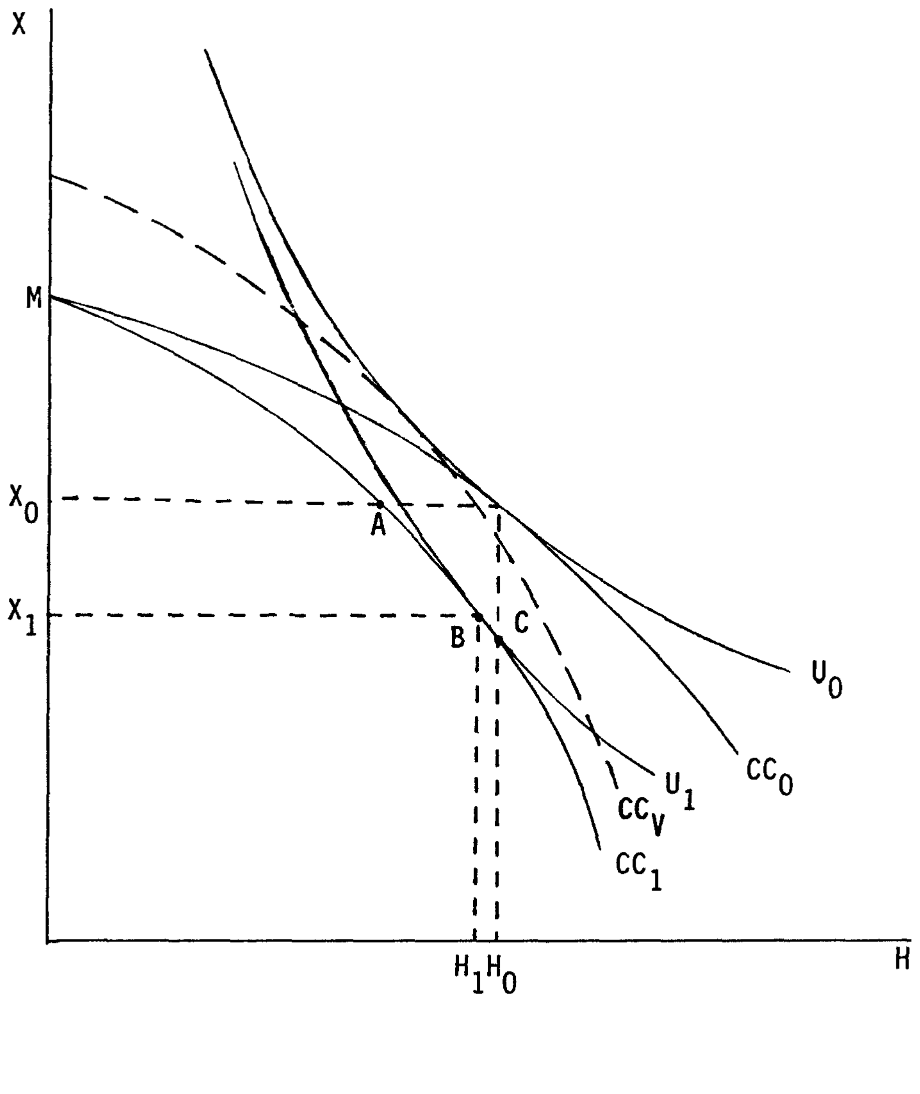

Figure 2.6 Compensating Variation ($\Delta X = |CV|$)

changes in water quality. The ΔX in this case approaches the maximum allowed by the assumption of $\Delta H=0$. This implies a large increase in defensive expenditures as a result of the increase in C. Thus, a large valuation of health is implied. After receiving a CV, decreasing health (by reducing defensive expenditures), and inferring a negative income elasticity for health, appears to contradict previous behavior. Thus, for a rational individual, the statement can be made with considerable confidence that $|CV|$ does not equal ΔD.

The two possibilities that remain are $|CV|>\Delta D$ or $|CV|<\Delta D$. Considering the previously stated possibilities along CC_1, the tangency of U_1 and CC_1 will occur either between points A and B or between points B and C in Figure 2.6. Upon receiving a CV, the rational expectation was that health would increase. Referring back to Figure 2.4, the increase in C causes the slope of CC_1 to be greater in absolute value than CC_0 at all vertical cross-sections so the point of tangency between CC_V and U_0 will be to the left of H_0 with the assumption of convex indifference curves. In the case in Figure 2.5, health will increase with a CV, with a resulting health level above H_1 after a CV.

Based on the assumption that health increases with an increase in income, such as a CV, and constant prices, the expected intersection of U_1 and CC_1 would occur between points A and B along CC_1 in Figure 2.6. Along this possibilities curve, $|CV|$ will always be greater than the change in defensive expenditures. So, ΔD is a lower bound estimate of $|CV|$ based on reasonable assumptions.

Some indication can be made as to the accuracy of ΔD in estimating $|CV|$. Note that in the situation where $\Delta X=|\Delta D|=0$ in Figure 2.4, the difference between $|CV|$ and ΔD was the entire $|CV|$ measure. As ΔX increases in absolute value, the difference between $|CV|$ and AD decreases until a situation such as Figure 2.6 occurs when $|CV|$=AD. Thus, as the absolute value of ΔX, or the value of ΔD, increases, AD is a more accurate measure of $|CV|$. The accuracy of ΔD in estimating $|CV|$ depends on the judgement as to the size of ΔD. A "large" ΔD implies an accurate estimation of $|CV|$ while a "small" ΔD means a poor estimate of $|CV|$. No standard exists for determining a large or small ΔD but various measures may be suggested, such as determining the percentage of households which increase D in response to an increase in C, comparing the ΔD with the level of D before the increase in C, determining relative income shares of averting expenditures and estimating income elasticities. Data collected from different communities could prove helpful in determining relative sizes of increases in defensive expenditures associated with increases in contamination levels.

The next step in the analysis is to compare the change in D with the EV. The restrictive assumptions made for the CV analysis can also be applied to the EV case. Assume that the increase in C will result in an optimized solution where X and H will not increase.

Proceeding similar to the CV analysis, define one extreme outcome as constant health after an increase in C, actual or proposed, without any EV considerations. This is represented in Figure 2.7. The change in defensive expenditures is (X_0-X_1).The slope of CC is greater in absolute value than CC_0 at any vertical cross-section. Thus, shifting CC_0 inward to establish an EV will result in a tangency with U_1 that must be to the right of a vertical line from H_0. Along the line H_0, EV is greater in absolute value than the change in defensive expenditures.

Next, the case of no change in X after an increase in C is depicted in Figure 2.8. Since X does not change, $\Delta D=0$. However, the absolute value of EV is greater than zero. The possibilities curve of optimized solutions of X and H after the C increase can now be defined as CC_1 between points A and B in Figure 2.8. Along this possibilities curve, the $|EV|$ will always be greater than the change in defensive expenditures. Therefore, ΔD will be a lower bound estimate of $EV|$.

Note that $|EV|$ is greater than ΔD in Figure 2.8 by the total amount of $|EV|$. As ΔD increases to an assumed maximum value in Figure 2.7, the difference between $|EV|$ and ΔD decreases. Similar to the CV case, the accuracy of ΔD in estimating $|EV|$ is a function of the size of ΔD, which is determined by the shapes of the indifference curves and consumption constraints. A difference between the CV and EV cases is that ΔD will never be equal to or greater than $|EV|$ since the EV curve must be tangent to U_1 to the right of where CC_1 is tangent to U_1.

The case of $|EV|=|CV|$ is illustrated in Figure 2.9. $|CV|$ will equal $EV|$ when the vertical distance between U_0 and U_1 is constant for all vertical cross-sections in the space being considered. In this graph, the EV measured along the vertical line at H_0 is equal to the $|CV|$, measured along a vertical line at H_1, when the vertical distance between U_0 and U_1 is the same at the two cross-sections. Note that this case implies that the health level that would be chosen after a CV is equal to H and the health level chosen after an EV is H_0. This can be interpreted by stating that health must have a zero income elasticity of demand for $|CV|$ to equal $|EV|$. As Freeman (1979) explains, if health has a positive income elasticity, the CV will exceed the EV in absolute value. Thus, if health is assumed to have a positive income elasticity, $\Delta D< EV <|CV|$ and ΔD would be a more accurate lower bound measure of $|EV|$ than CV .

2.10. Possible Complications and Further Considerations of the Model

The applicability of any economic model depends on the reasonableness of the assumptions. An important assumption considered by Bartik (1988) was that defensive measures provide no sunk costs. This assumption is valid for all averting actions related to home water use except the installation of a home water purification system. Such a system is likely to provide benefits lasting beyond the duration of a specific water contamination incident. Violation of this assumption implies that the full purchase price of the durable good can not be attributed to the change in water quality. Some proportion of the cost of the system may be attributed to the expected protection to be provided after a contamination incident. An important distinction needs to be made between actions which are a

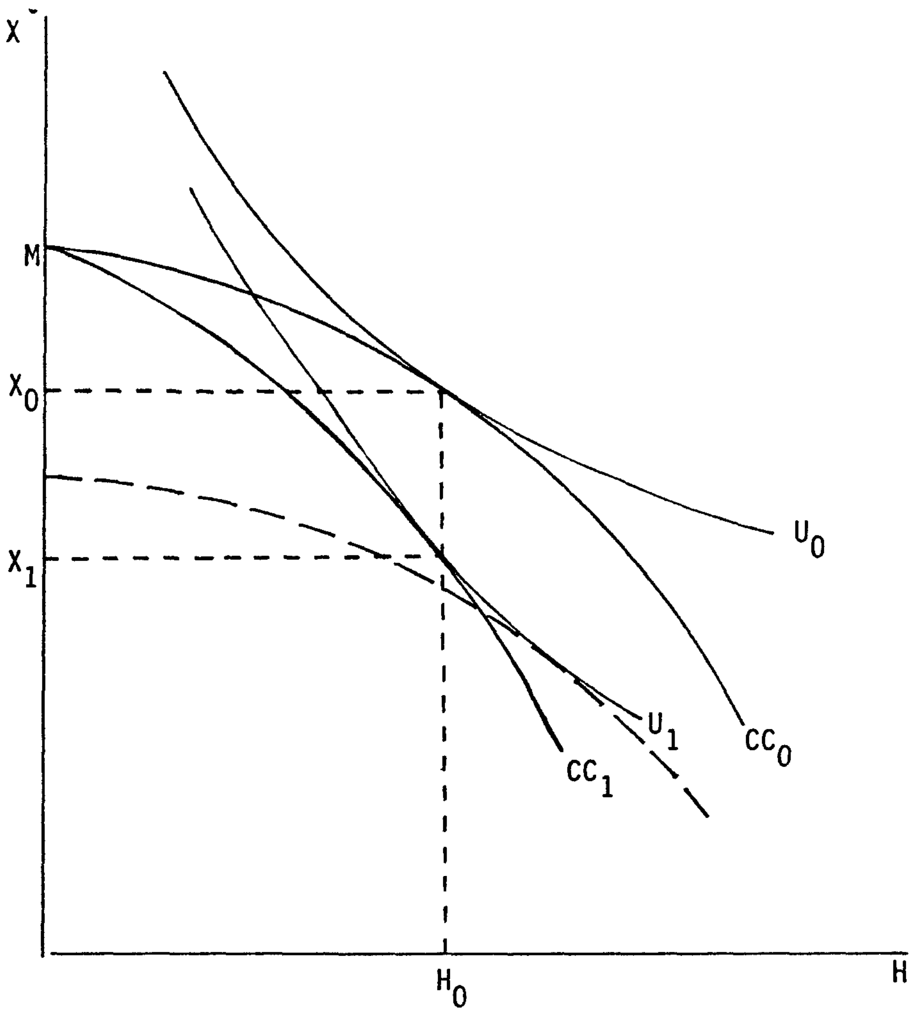

Figure 2.7 Equivalent Variation ($\Delta H=0$)

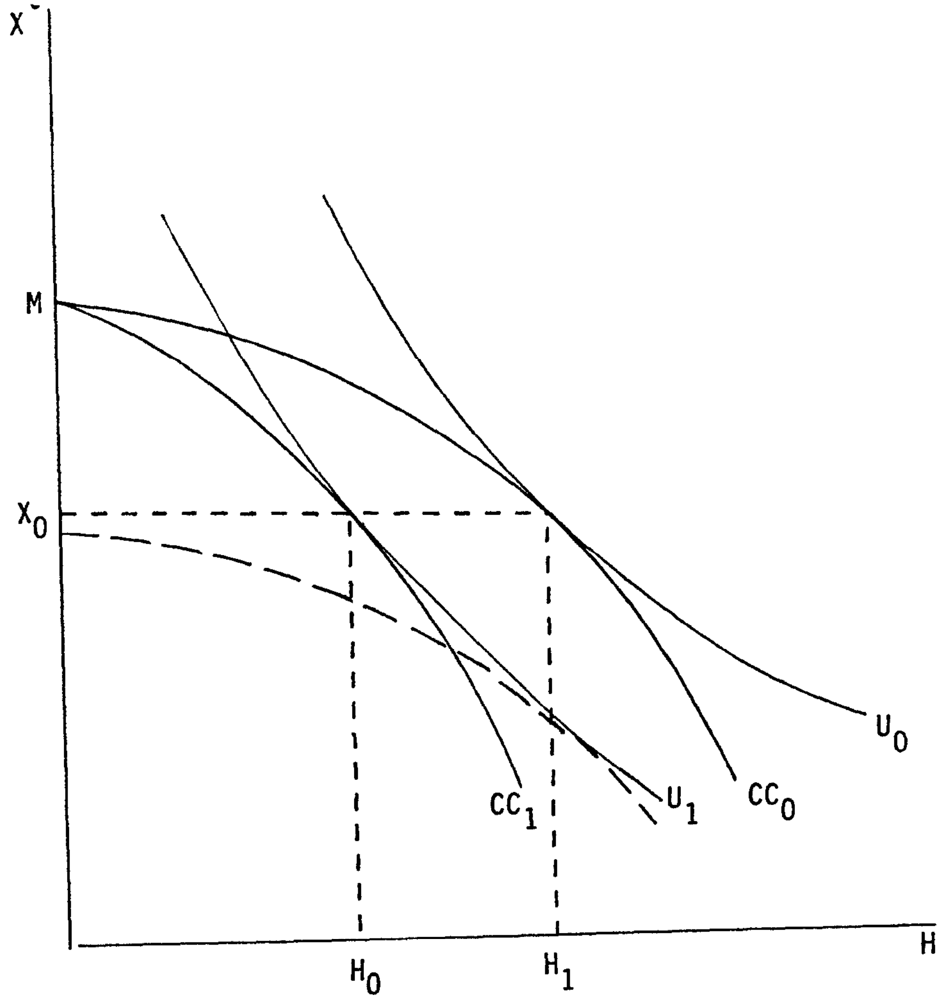

Figure 2.8 Equivalent Variation ($\Delta X=0$)

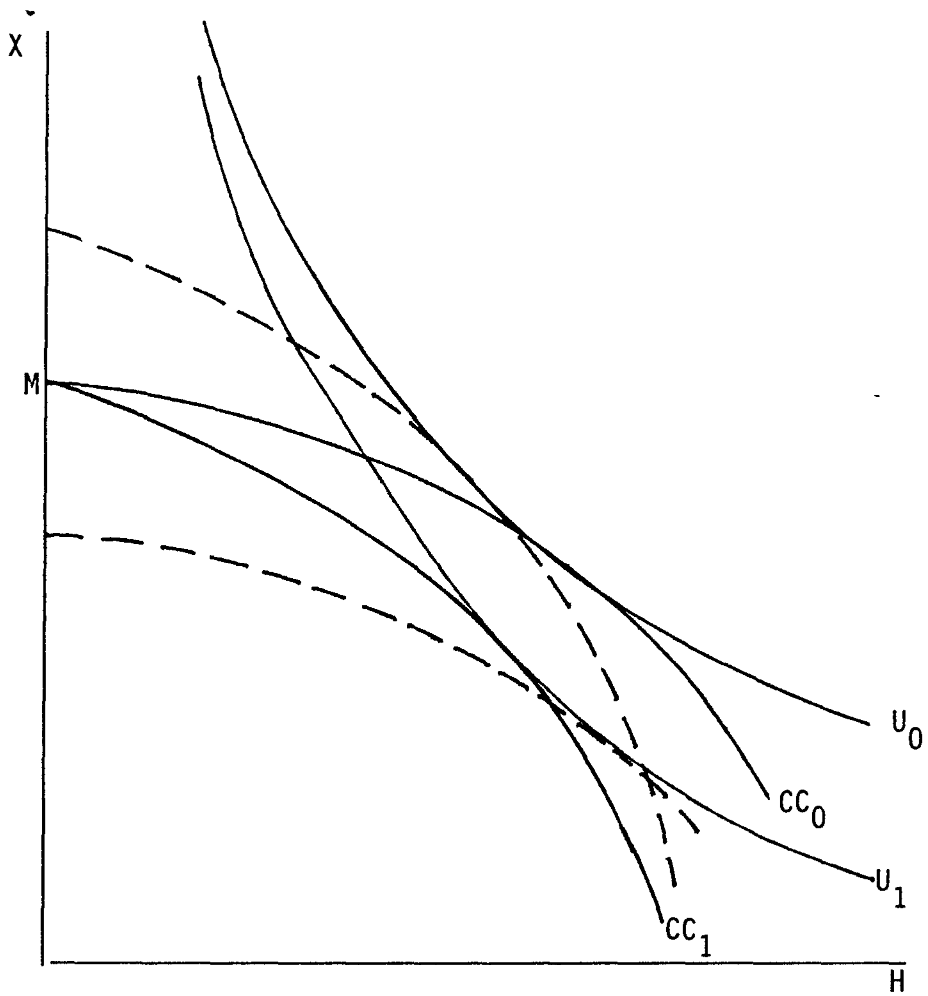

Figure 2.9 Compensating Variation = Equivalent Variation

direct response to the change in water quality and those which are taken because of other reasons. However, such a distinction is not always clearly defined. Some doubt may exist concerning whether some costs should be included as averting expenditures. When such doubts arise, these costs should not be included as averting expenditures since a lower-bound estimate is being calculated. Averting expenditure estimates may no longer be a lower-bound estimate of WTP if questionable costs are included.

Courant and Porter (1981) indicate that defensive expenditures provide poor estimates of WTP when the quality of the environmental factor enters directly into the utility function. The model presented here assumes that C does not enter into the utility function. However, this may not always be true for household water quality issues. Situations where water quality directly enters into the utility function include a contamination incident or defensive measure affecting the taste or odor of the water and situations involving aesthetic factors of water quality.

Another important assumption to consider is that the averting goods must exhibit no joint production (Pollack and Wachter, 1975). Jointness in production implies that the averting good is an input into the production function of two or more commodities. If an averting good exhibits jointness in production, expenditures on the good are theoretically divided among the production of each commodity involved. A good example of jointness in production is the use of air conditioners to reduce exposure to air pollution. A portion of air conditioner expenditures may reflect a desire to protect oneself from the harmful effects of air pollution. However, another portion may reflect a desire to cool one's personal environment. Thus, air conditioners may be an input into the production of two commodities, health and personal comfort.

Whether or not averting goods affecting water quality exhibit jointness in production seems to depend on the presence of contamination. Consider the purchasers of bottled water in a community which has not experienced a recent contamination incident. The reasons for purchasing bottled water are likely to include protection from hazardous chemicals and better taste than tap water. In this situation, all bottled water expenditures can not be directly attributed to existing contamination since the taste may also be a reason for purchasing bottled water. Next, consider bottled water purchases in a community which has experienced a recent contamination incident. The expenditures before the contamination are likely to exhibit jointness in production for the same reasons given above. However, an increase in bottled water expenditures after the contamination appears less likely to exhibit jointness in production. The increase in bottled water expenditures can be directly attributed to the increased health risks of the contamination and not changes in tastes and preferences.

Several other possible complications of defensive expenditure models, noted by Barrington and Portney (1987), arise when income effects are considered due to work loss through illness caused by the contaminant. The cost of illness due to a contamination incident is most often applied to contaminants which cause short-term health effects, such as the Giardia contamination incident studied by Harrington, et al. (1985). The health related effects of the contamination incident under study are long-term and not easily attributable to the specific contaminant. Thus, the cost of illness due to the contamination incident cannot be directly estimated. However, a method may be proposed to estimate the cost of illness through an indirect statistical approach when the health effects are long-term.

Consider a contamination incident which effects a population size of P. Assume that the contaminant causes no short-term health effects but increases the probability of contracting a certain disease or ailment, such as cancer. Scientific analysis can determine the increased probability, I, of developing cancer associated with the duration of the contamination incident. Thus, (P*I) is a statistical estimation of the number of cancer cases in the future in the affected area that can be attributed to the contamination incident. An estimation of the cost associated with an occurrence of cancer must be made. This figure could be discounted by determining the expected time until onset of cancer and applying a discount factor. Little information exists concerning the long-term health effects of many contaminants. This research will not attempt to estimate the cost of illness associated with a contaminant with long-term health effects but possible cost of illness factors should be kept in mind when referring to the estimates presented.

2.11. Risk Perception and Valuation

Individuals are exposed to many environmental factors that present health risks. Since evidence of averting behavior does exist, individuals appear to perceive certain risks to be significant enough to undertake protective actions to reduce exposure. The actual health effects produced by changes in pollution levels are not completely understood in medical terms and health risks are understood even less by the public (Runge, 1983). Also, there is likely to be differences between actual and perceived risk levels (Smith, 1985) due to a lack of information or attitudes.

Weinstein and Quinn (1983) gave several examples of policies which they believed to be inefficient, including the Clean Air Act and amendments to the Social Security Act. They attributed these inefficiencies to inadequate measurement of risk and the value of statistical lives saved by policies. The suggestion was made that voluntary risks are accepted more willingly than involuntary risks so directly comparing different types of risks may be improper. Also, policies which reduce risks with certainty may be valued more than those which have a degree of uncertainty. Also, anxiety seems to have a large role in determining views towards environmental risks but this may not be empirically measurable.

Smith and Desvouges (1988) reported that ex ante expenditure functions can be utilized to incorporate risk attitudes in valuing policies which reduce risk. Further, risk perceptions may not necessarily correspond to actual risk levels since individuals make decisions based on their perceptions and attitudes. They indicated that additional information needs to be elicited concerning individuals' risk perceptions. The lack of empirical data on risk perceptions and attitudes also affects the theoretical development of risk issues. Smith and Desvouges indicated that theoretical and empirical work should be conducted in conjunction with each other to obtain results which have policy relevance.

Another recent approach to risk analysis is conducting economic experiments, such as Shogren (1988). In an experimental setting, respondents have the possibility to receive actual monetary compensation, based on their choices during a personal interview. Shogren stressed the beneficial effect that instruction, given by the interviewer, had on the effectiveness of respondents' choices. However, difficulties may arise in applying experimental methods of valuing environmental risks since the consequences of environmental decisions may occur over a long period of time. These methods are presently used to resolve immediate risk choices.

The two methods most commonly used to value risk changes are human capital and the "value of a statistical life" (Viscusi, 1986). The human capital approach normally uses labor market information to determine risk and income differences of different occupations. Workers are assumed to be compensated (in terms of higher wages) for undertaking high risk jobs. Another application of the human capital approach is to value lost lives by foregone wages. The value of a statistical life represents society's willingness to pay for actions that reduce expected deaths by one without reference to the specific individual whose life is saved (Fisher, et al., 1989) . For example, suppose a policy is anticipated that reduces a certain risk of death by 1 in 1,000. Assume 1,000 individuals were willing to pay an average of $100 each for this risk reduction. Since the policy reduces the risk by 1 in 1,000, one death in a community of 1,000 individuals is likely to be prevented by the policy. Note that this represents a statistical projection, not actual health effects. Thus, the value of a statistical life is calculated as society's willingness to pay to save this one statistical life, or $100,000.

The value of a statistical life approach represents an improvement over the human capital approach because the values estimate the actual benefits (costs) of health increases (reductions) (Viscusi, 1986). Society's willingness to pay to reduce risks using the value of statistical lives has recently become an important instrument in designing public policies which influence risk levels (Fisher, et al.,1989). The pertinent factor in designing environmental risk policies should be the reduction in the probability of death or injury for a large number of individuals (Viscusi, 1986). This emphasizes the need to estimate the value of a statistical life, which may vary according to the type of risk (Thompson, 1986) . Viscussi reasoned that the value of a statistical life should also vary according to the baseline risk. He illustrated this point by referring to a Russian roulette example. One would be willing to pay an "infinite" amount for removal of the first bullet but a lower price for the removal of successive bullets.

Many researchers have empirically attached values to a statistical life by using various means to elicit information of WTP for risk reduction. Contingent valuation methods have frequently been used (Thompson, 1986; Smith and Desvouges, 1987; Viscusi, Magat and Huber, 1989). Smith and Desvouges surveyed residents of suburban Boston by asking direct willingness to pay questions for different probabilities of risk reduction from different baseline risks. Their findings contradicted conventional utility theory by indicating that the marginal valuation of risk decreased with increases in baseline risk. They hypothesized that this unexpected result may be attributed to respondents' lack of understanding of actual risk levels. They suggested that future work concentrate on better elicitation of risk perceptions and behavioral responses to actual incidents in which risk changes occur.

Viscusi, et al. (1989) offered a new approach which used CV methods to measure risk-risk tradeoffs. Indifference points were determined by presenting two areas, one with a fixed probability of auto death and the other with a comparatively low risk of chronic bronchitis. The risk of developing chronic bronchitis was increased until the respondent indicated indifference between the two locations.

The results of several estimation attempts to value a statistical life were summarized by Viscusi (1986). Table 2.1 lists several of the research attempts included in the article. The values ranged from $560,000 to $11 million, measured in 1982 dollars. He emphasized that the lower values tended to reflect valuations towards voluntary risks, such as seat belt use, while involuntary risks such as those associated with nuclear power and airline safety, generally resulted in larger values. Viscusi stated that with better understanding and research, policy analysis should be able to catalog acceptable and reliable values for statistical lives in a variety of risk situations.

2.12. Summary

This chapter presented a model of defensive expenditures for marginal and non-marginal changes in the quality of environmental factors. The model indicated that defensive expenditures are a lower bound estimate of theoretical measures (WTP) under certain assumptions. Most of the model's assumptions appear to hold true when considering changes in household water quality. Therefore, the change in defensive expenditures associated with a contamination incident can be used to generate a lower bound estimate of the costs of the incident.

Risk perceptions may influence individuals' behavior (including averting expenditures). Finally, the "value of a statistical life" has become an important instrument to value risk changes in policy analyses. However, no previous studies have calculated the value of a statistical life using information on averting expenditures.

Table 2.1 Summary of Obatined Values of a Statistical Life (Viscusi, 1986)

Investigator	Sample	Implicit Value of Life
1. Blomquist (1980)	Seatbelt usage, Panel Study of Income Dynamics, 1972	$560,000
2. Brown (1980)	National Longitudinal Survey, 1967-1973	$1-$1.5 million
3. Olson (1981)	Current Population Survey, 1973	$7.4 million
4. Portney (1981)	Air Pollution and Property Values	$593,000-$890,000
5. Thaler and Rosen (1976)	Survey of Economic Opportunity, 1967	$580,000
6. Viscusi (1979)	Survey of Working Conditions, 1970-71	$2.9-$3.9 million
7. Viscusi (1981)	Panel Study of Income Dynamics, 1976	$7-$11 million